

CHAPTER 2

The Basis for Lake and Reservoir Nutrient Criteria

- A. Historical Perspective
- B. The Nutrient Paradigm
- C. Connecting Watershed Loading to the Lake:
A Mass Balance Model Approach
- D. Trophic State Classification Systems
- E. Uses of Trophic State Indices

A. Historical Perspective

Large-scale comparative studies of lakes have enabled scientists to identify key variables that influence lake structure and processes (Peters, 1986). This empirical approach has its roots in regional studies (e.g., Naumann (1929) and Thieneman (1921)) and in early among-lake comparisons of lake function such as the role of morphometry on lake productivity (Rawson, 1955) and nutrient input regulation of lake fertility (Edmondson, 1961).

The eutrophication process was quantified by Vollenweider (1968, 1975, 1976), which brought the large-scale comparative approach to the forefront of limnology. Vollenweider developed a mass balance model using literature data from a diverse population of temperate lakes to demonstrate a surprisingly strong relationship between nutrient inputs to lakes and concentration of nutrients within the lake. This relationship was sufficiently powerful to stand out against other sources of among-lake variation and signaled that nutrient loading, as modified by hydrology, morphology, and in-lake sedimentation, was the dominant factor explaining lake eutrophication.

The element phosphorus was the focus of study because overwhelming evidence suggested that phosphorus limited algal growth in many aquatic systems (Schindler, 1977). Phosphorus values were highly correlated with algal biomass in lakes (Sakamoto, 1966); in turn, water clarity was shown to vary with algal levels (Edmondson, 1972). With these linkages quantified, the science of lake management arose around the premise that reductions in nutrient loads would reverse eutrophication, as measured by reduced nutrient concentrations, algal levels, and greater water clarity.

Empirical models provided limnologists with a quantitative basis for estimating the level of response to be expected from a given change in nutrient load from point and nonpoint sources. Models were the tools for forecasting the capacity of a lake to withstand change in its trophic state with various degrees of human development in its catchment (Dillon and Rigler, 1975). Recently, land use, as a surrogate for external nutrient loading, was used to effectively predict algal chlorophyll in lakes (Meeuwig and Peters, 1996); the strength of this approach stems from the strong correlation between nutrient losses and land use practices in catchments (Smart et al., 1985). This linkage of land use to chlorophyll, a widely accepted measure of lake trophic state, is additional evidence for the importance of external control on lake processes.

The large-scale comparative approach placed individual lakes within a continuum, from least to most fertile. With this understanding, lakes lost some of their individuality because scientists now viewed them within the context of this continuum. The functional relations between external nutrient loading, algal biomass, and water clarity were summarized in a small number of general models. These models were typically based on regression analyses of data from individual lakes, averaged over a sampling

season. The models quantified large-scale lake functions and provided the conceptual basis for lake management and restoration.

Because these early data were drawn from a diverse group of lakes, both in terms of lake type and geographic location, these models often are referred to as “global models.” An underlying assumption was that processes responsible for the large cross-sectional patterns in these global relations also operate within single systems over time (Prairie and Marshall, 1995). About the time of Vollenweider’s work, Edmondson (1972) applied virtually the same concepts to data covering the enrichment and recovery of Lake Washington. Edmondson’s work was tangible confirmation that a single lake responded to nutrient loading, as the pattern drawn from the data of many lakes would suggest. The remarkable feature during the 1970’s was that a quantitative paradigm for lake function had been proposed based largely on data drawn from the literature. It was a synthesis of ideas from earlier descriptive and empirical studies. A feature of predictions from empirical models was that there was a great uncertainty in them; many models exhibited an order of magnitude variation. This variation was a point of concern and the focus of subsequent study.

During the two decades since the empirical period of the mid-1970’s, lake management has been influenced by several major thrusts that have modified, but not invalidated, the work of that period. With expanded data sets over the past 20 years, the original global generalizations have been modified showing that in highly enriched lakes algal biomass does not increase in a uniformly linear relationship to phosphorus in all lakes (McCauley et al., 1989; Prairie et al., 1989; Watson et al., 1992) because other environmental factors also play a role. The Organization for Economic Cooperation and Development (OECD, 1982) project was an early effort to systematically gather data and quantify the relationship between nutrient load in waters and their trophic reaction. This project, composed of four regional studies (Alpine, Nordic, Reservoir and Shallow Lakes, and North American), attempted to corroborate Vollenweider’s generalizations. Its approach shifted the focus of among-lake comparisons from a global scale to studies within regions and studies of specific lake types.

Since then, several regional studies have used the comparative approach to generate empirical models specifically for local conditions. These regional studies have demonstrated the importance of other factors regulating algal biomass in lakes. Four other factors include nitrogen (Canfield, 1983; Pridmore, 1985), light limitation due to suspended solids (Hoyer and Jones, 1983; Jones and Knowlton, 1993), lake morphometry (Riley and Prepas, 1985), and grazing by herbivores (Quiros, 1990).

B. The Nutrient Paradigm

The concept of nutrient criteria is based on the idea that nutrients produce changes in lakes and reservoirs that are considered to be detrimental to the function or use of the water body. This idea of nutrient control of water body function is not new; it can be traced back to when Einar Naumann, the Swedish limnologist, elucidated the major part of the nutrient paradigm in 1929. His ideas of the relationship between nutrients and lakes can be summarized in the following four statements:

- The primary factors that determine algal biomass (the amount of plant organic material) are the plant nutrients phosphorus and nitrogen.
- The geology (and land use) within the lake’s watershed determines the amount of nutrients that enter the lake and, therefore, plant biomass.

- Changes in the plant biomass affect the entire lake's biology.
- There is a natural ontogeny to lakes; the amount of plant biomass and, therefore, the entire biology of the lake increases as the lake ages.

Although there have been many significant additions and improvements in our understanding of lakes since Naumann, his original concept of nutrients remains the basis of the nutrient paradigm. Below, each statement is examined as it refers to the need for and the development of nutrient criteria for lakes.

1. Phosphorus and Nitrogen as Limiting Factors for Algal Biomass

The primary factors that determine algal biomass (production) are the plant nutrients phosphorus and nitrogen. When Naumann (1929) suggested this concept, he was probably drawing on a much older concept, Justus von Leibig's Law of the Minimum. The law, as it is formulated today, states that the factor that is in shortest supply relative to the needs of the plants limits the growth of those plants. The concept is central to the nutrient paradigm in lakes because it insists that very few factors (usually only one factor, often a plant nutrient such as nitrogen or phosphorus) will actually limit plant growth.

If only one factor, such as phosphorus, was always limiting, the task of developing nutrient criteria would be a simple matter of determining limits on that single factor. Unfortunately, the factor that limits plant biomass may (1) change seasonally or over longer periods of time, (2) vary depending on the land use, or (3) vary regionally. It would make little sense to construct a single nutrient criterion when that nutrient may not necessarily limit a target lake or lakes. It is for that reason that the emphasis of this document is the development of nutrient criteria based on both the nutrient inputs and the biological response.

The causal variables such as phosphorus and nitrogen are essential criteria because they will be the limits necessary to establish management objectives and are usually directly related to discharge runoff abatement efforts. Although phosphorus is the limiting factor for most lakes and reservoirs, in some regions the nutrient paradigm centers on nitrogen rather than phosphorus, especially where sewage treatment plant effluent is involved (Canfield, 1983; Pridmore, 1985; Jones et al., 1989). These regions are often in the subtropics or at high latitude or altitude (Wurtsbaugh et al., 1985; Morris and Lewis, 1988) but are also found in parts of Britain. In these lakes, nitrogen rather than phosphorus explains the among-lake variance in algal chlorophyll, and chlorophyll-total nitrogen regressions match the "fit and form" of chlorophyll-total phosphorus regressions developed for phosphorus-limited temperate lakes. The reason for nitrogen limitation is not yet understood because of a long-held tenet in limnology that states that nitrogen fixation will compensate for shortfalls (Schindler, 1977) and that nitrogen limitation is not a persistent condition. This belief does not seem as universal as once thought (Knowlton and Jones, 1996). In some regions, nitrogen limitation may be a function of abundant phosphorus in the geological formation of the region (Canfield, 1983).

Nitrogen limitation also may be tied to efficient nitrogen cycling in subtropical forests or may be a function of nitrogen uptake by rice and other crops in the subtropics. In high-elevation lakes, phosphorus may be contributed by soil weathering, whereas nitrogen is rare in these low organic soils. A recent literature review showed that nitrogen limitation was about as common as phosphorus limitation (Elser et al., 1990). Detailed water chemistry data from the midwestern lakes suggest that nitrogen values in the epilimnion fall during summer but that phosphorus values remain more constant. These data suggest phosphorus may be cycled more efficiently than nitrogen and that without external inputs, late summer nitrogen limitation can be expected. These results do not imply that continued focus on phosphorus for

eutrophication control is unwarranted; however, a better understanding of the frequency and extent of nitrogen limitation is needed to discern lake function. Nitrogen criteria as well as phosphorus criteria are appropriate.

Response variables such as chlorophyll *a* and algal or macrophyte species or biomass indicate the relative success of the nutrient management effort. By carefully incorporating both the causal and response elements, a State or Tribe should be able to fine-tune its criteria to meet the necessary enrichment levels for a given class of lakes. These variables are described in more detail in Chapter 5.

2. Role of the Watershed

The geology (and land use) within a lake's watershed determines the amount of nutrients that enter the lake and, therefore, plant biomass. This statement is probably the primary reason for the development of nutrient criteria: human activity in the watershed affects a lake's function. It is the reason behind the National Nutrient Assessment Workshop's conclusion that changes in land use can serve as an early warning system for changes in lakes (U.S. EPA, 1996).

In simplest terms, a lake's nutrient concentration is affected primarily by the rate of weathering and erosion from the soils in the watershed. If the underlying geological structure is granitic, then the rates of weathering will be low and both the productivity of the terrestrial vegetation and the concentration of nutrients in the runoff from the watershed will be low. On the other hand, if the underlying bedrock is sedimentary, the weathering rates will be higher and the fertility of the soil and the nutrient content of the runoff water will be higher as well. Consequently, Naumann (1929) observed that lakes in regions of sedimentary rock had higher algal densities (were greener) than lakes in granite-based watersheds. (For the purpose of this manual, atmospheric deposition of nitrogen and phosphorus, while possibly important, is accepted as a regional constant subject to further attention as our management technology improves.)

Human activity has at least two effects on the natural load of nutrient input to lakes: (1) it disturbs the overlying vegetation, exposing the soil to increased weathering and erosion, and (2) it adds easily erodible nutrient-containing material, such as fertilizers and animal waste, into the watershed. As the biological surface of an undisturbed watershed is disrupted, and as people move into the watershed, it can be expected that there will be increased soil and nutrient runoff.

Of course the degree of disturbance relative to the size of the lake will affect the impact of the disturbance; building a summer cottage would not have the same impact on a lake as would clear-cutting a forest or developing a condominium complex. Sometimes the term "assimilative capacity" is used to imply that the lake has a certain capacity to absorb the impact of disturbance. This concept, although comforting, probably has little basis in fact. Impact, until demonstrated otherwise, is probably better thought of as a continuous response to nutrient increases. The degree of change will depend on other factors, such as the size of the lake, and the change may not be immediately or even ever detectable to humans or their monitoring instruments. However, whether detected or not, changes do occur. It is for this reason that watershed disturbance is a sensitive early warning of lake change. Clearly, biological impact within the lake will be directly related to the increased amount of nutrient loading, and that impact will occur, whether or not it is detected.

Naumann (1929), nonetheless, used the relationship between nutrients and plants to establish a trophic state classification. He probably began his classification scheme with the perfectly reasonable goal of classifying lakes into those with low (oligotrophic) and high (eutrophic) plant biomass.

Oligotrophic lakes were clear with little algae, whereas eutrophic lakes were green. He then added to his classification system the causal factors that produced this degree of greenness, for example, the amounts of nitrogen or phosphorus. He called these the “factors of production.” Oligotrophic lakes were those that had low production (biomass) because they were low in nutrient concentrations. Eutrophic lakes were green because there were abundant nutrients to support the growth of algae.

The combination of the factors that affected production (causal factors) with plant production itself (response variable) allowed for a suite of trophic classes that dissected lakes into groups of varying production based on the factor or factors that were thought to limit that productivity. The classic oligotrophy-eutrophy axis was based on limitation by nutrients. The mesotrophic category was added to describe situations intermediate between oligotrophy and eutrophy. The term “hypereutrophic,” or hypertrophic, was added by Wetzel (1966) to describe situations of extreme eutrophy where light, not nutrients, is the dominant environmental factor controlling growth. This continuum of trophic states is illustrated in Figure 2.1.

3. Trophic Causal Chain

Naumann (1929) was very insightful in recognizing that the components of the lake are an interconnected system; as one component—the plants—responds to nutrient inputs, other biological, chemical, and even physical components would be affected as well. Increases in nutrient loading do not necessarily directly affect any component other than the plants, but by various pathways, other components of the lake ecosystem, such as zooplankton, fish, and hypolimnetic oxygen concentration, are affected as well. This trophic state cascade is depicted in Figure 2.2.

Because of these linkages between components, numerous variables may respond to varying degrees to increases in nutrients. Not only will algae or macrophytes increase, but zooplankton and fish biomass may increase as well, plant and animal species may change (with some going extinct), and hypolimnetic oxygen may be depleted. People react to the various changes or symptoms of lake condition reflected in the chemistry and biology that cascade from the change in loading, not directly to the change in loading or nutrient concentration itself.

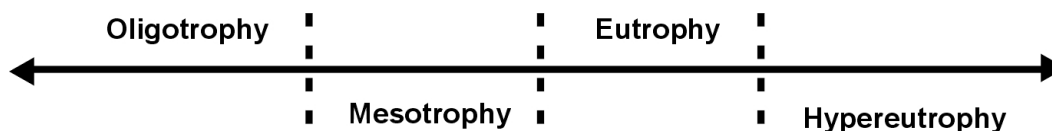


Figure 2.1. A trophic continuum.

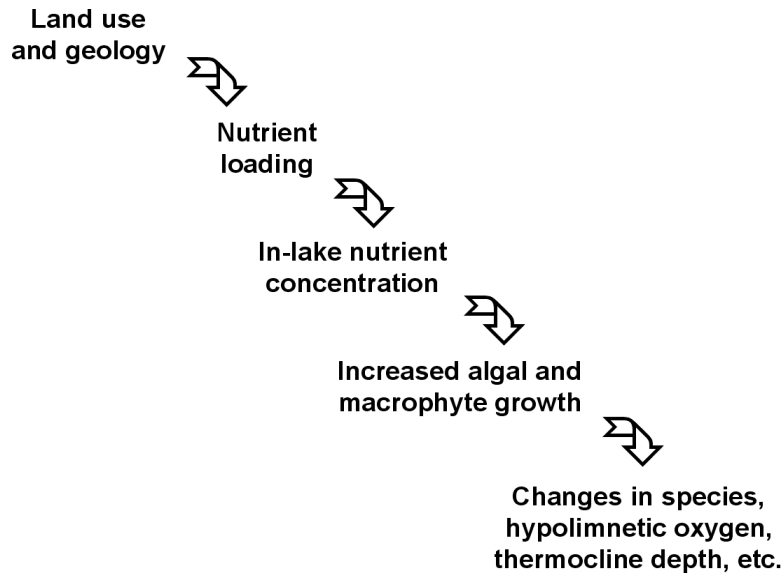


Figure 2.2. The trophic causal chain.

This cascade of biological and chemical changes produces a variety of choices for the response variables needed to supplement the causal variables in the formulation of nutrient criteria. Choice of the response variable can be made based on sensitivity to change, cost of measurement and analysis, or importance to designated use.

4. Lake Aging

The idea that lakes undergo directional change in plant production as they age was probably related to the observation that shallower lakes appeared to have more plant biomass in them than deeper lakes. This observation later translated into the idea that increases in plant biomass were inevitable as a lake ages and fills in. This concept has led to terms such as “natural eutrophication” to describe inevitable increases in plant biomass as a lake becomes older and shallower. If natural eutrophication is thought to proceed at a rate related to inputs from the watershed, then we might expect to see accelerated rates of eutrophication if cultural influences occur in the watershed (cultural eutrophication).

If trophic state is a description of the biological condition of the lake, eutrophication describes a lake that is becoming more eutrophic (Figure 2.3). Specifically, it describes a change in the direction of eutrophy. A lake does not have to become eutrophic to have undergone eutrophication; it only has to move in the direction of eutrophy. Oligotrophication describes the process of a lake moving in the alternative direction, towards oligotrophy.

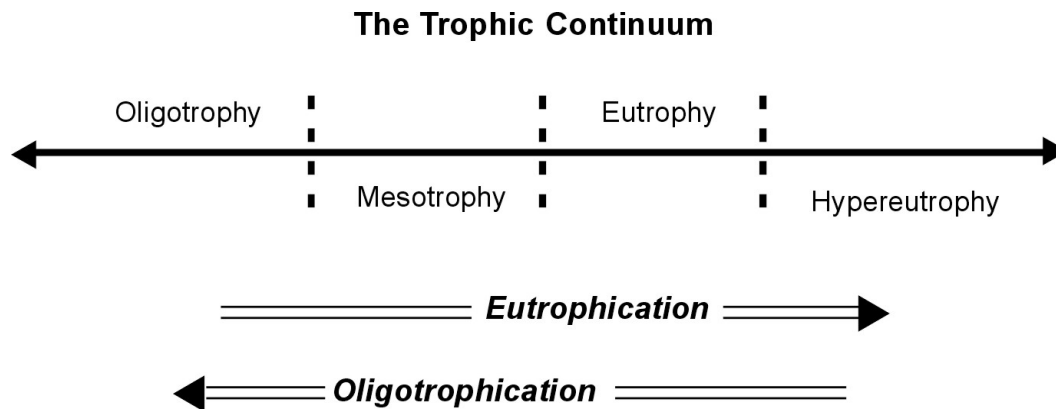


Figure 2.3. Eutrophication and oligotrophication in relation to the trophic continuum.

The term “nutrient enrichment” is in many, if not most, instances an alternative term for eutrophication. However, the emphasis in that term is on the increase in nutrients rather than on the lake’s response to that enrichment. If causal factors such as nutrient enrichment are closely linked via our terminology with the lake response, problems can arise if, for example, a lake’s biology changes without any change in nutrient loading or, conversely, if nutrient loading occurred without a coincident change in biology. These are not just hypothetical situations. The addition or removal of a benthivorous fish, such as bullhead or carp, can change the internal regeneration of nutrients and change the biological condition of the lake without any enrichment, or at least external enrichment, of the lake. The increase in grazing on algae because of the addition of a piscivore or the removal of the zooplanktivores also can alter the amount of plant biomass without the need of an alteration of nutrient loading. These manipulations, often called biomanipulations (Shapiro et al., 1975), are a type of lake manipulation that can alter the state of the lake without a change in nutrient load.

Natural eutrophication should not be confused with naturally eutrophic. The latter term describes lakes in watersheds where the natural load of nutrients is high despite the absence of human activity. Natural eutrophication describes a belief that lakes, presumably all lakes, have more plant material within them (become more eutrophic) as a natural part of becoming older.

The concept of natural eutrophication is probably correct to the extent that processes within the lake such as nutrient regeneration may enhance the effect of inputs from the watershed. The importance of these internal processes are still not well understood, especially along a gradient of lake aging. If becoming shallower were the only consideration, then the internal concentration of nutrient would increase only to a level as high as that in the incoming water. A lake in a watershed in which the concentrations of nutrients were very low in the incoming water would not become that much more eutrophic even if it did become shallower. However, a shallower lake also may have increased macrophyte growth and increased regeneration of nutrients from the sediments. In this case, the biological response would become increasingly independent of the external supply of nutrients.

A difficulty with dwelling on the possibility of natural eutrophication is that it emphasizes an inevitability of the eutrophication process; it also takes the focus off the immediate effects of the

watershed on the lake. Natural eutrophication is a process that is measured in terms of thousands of years, whereas the problems we encounter most often with lakes are the effects of processes that take only a few years to develop. Watershed disturbance can rapidly move a lake to a new level of nutrient concentration and biological response. More important, in most or many lakes, that response is, to some extent, reversible; we have not just moved rapidly down an irreversible path. Cultural eutrophication is, in fact, a reversible process, and nutrient criteria are an important element in this reversal.

C. Connecting Watershed Loading to the Lake: A Mass Balance Model Approach

Like many earlier nutrient loading models, Vollenweider used a mass balance model for the basis of the prediction (see Chapter 9 Section B). Below is a basic review of Vollenweider's model and how it is used to link loading to concentration in a lake or reservoir.

The term "mass balance model" comes from the assumption that a substance such as phosphorus cannot just appear or disappear from a reservoir; it must come from somewhere and it must go somewhere. The phosphorus going into the reservoir must either go out again through some outflow, be sedimented to the bottom, incorporated into macrophyte biomass, or remain in the water in either dissolved or particulate forms. It is this phosphorus that remains in the water that is of interest because it is the amount that is available for algal growth.

Mass balance modeling is done in a manner similar to keeping a checking account. The total amount of phosphorus entering the reservoir (loading) each year is measured. Loading describes the total amount of material being moved in a stream in a given amount of time. The loading from any source (i) is calculated as:

$$\text{Loading } (J_i) = \text{Water Discharge } (Q_i) \times \text{Concentration } (C_i)$$

or

$$J_i = Q_i C_i$$

The external loading, symbolized by Vollenweider by the letter J , can be calculated as the sum of the loading from all the external sources (i) to the lake:

$$J = \sum_{i=1}^n Q_i C_i$$

Loading often is used to measure export of a nutrient or sediment from a watershed. For example, it is important to gauge the effects of farming practices on erosion, so we might calculate the tons of sediment removed from a watershed over a year. On the other hand, if the stream enters a lake or reservoir, we might want to know how much material is entering that body of water. In this case, nutrient loading might affect water quality, and sediment loading might affect the fill-in rate.

Predicting the internal concentration of a substance in a lake is also done using a mass balance equation. The appropriate mass balance for this prediction is based on the idea that the rate of change of the total amount of a material (M) in a lake (dM/dt) is dependent on the total amount of material that enters a lake (J) and the total amount that leaves in the same time period:

$$dM/dt = (\sum Q_i C_i) - Q_o C_o$$

or

$$dM/dt = J - Q_o C_o$$

If the reservoir is not rapidly changing from one year to the next, the amount coming in one year should be equal to the amount going out:

$$\text{Inflow} = \text{Outflow} \quad (\text{i.e., } dM/dt = 0)$$

or

$$J = Q_o C_o$$

If we assume, as did Vollenweider, that the lake is completely mixed, then the lake concentration is equal to the outflow concentration:

$$C_{\text{lake}} = C_o$$

Therefore, the amount entering the lake will be equal to the amount leaving:

$$J = Q_o C_{\text{lake}}$$

Rearranging this equation, we obtain an equation predicting the concentration in the lake based on the external loading of the substance and the outflowing discharge of water:

$$C_{\text{lake}} = J/Q_o$$

Notice that the term (J/Q_o) has the units of concentration. Consider that it is the incoming loading divided by the outgoing water discharge.

This model is designed to predict the concentration of any conservative material. A conservative material, such as chloride or sodium, does not sediment within the lake basin, and the amount leaving the basin over the outflow should be equal to the amount entering. Conservative materials are not very interesting in themselves, but they are used as indicators of the accuracy of budgets of materials that do sediment within the lake. If the input of a conservative element is not equal to the output, then some other source of water and/or material has been neglected.

A nonconservative material, such as phosphorus, is one that is lost from the water column (e.g., sedimentation) within a lake basin. Because some material is lost from the water column, the input loading is not equal to the output loading. To model a nonconservative material, a sedimentation term must be added to the equation:

$$\text{Input} = \text{Output} + \text{Sedimentation}$$

Sedimentation was considered by Vollenweider to be proportional to the mass of the substance in the lake (M). The total amount of material in the lake (M) is calculated as:

$$M = C_{\text{lake}} V$$

where V = volume of the lake (m^3)

Vollenweider considered the amount of material lost to the sediments. This is designated by sM , where s is a first order fractional loss of the mass settled per unit time ($1/t$) and M is the mass of substance in lake ($C_{\text{lake}} V$). The sedimentation coefficient, s , is really a net sedimentation term, because the material may not only settle out of the water column but also may be resuspended into the column from the sediments.

The mass balance equation, with the added sedimentation term, becomes

$$dM/dt = J - Q_o C_o - sM$$

Vollenweider then assumed that over a calendar year the system would be near or at steady state, and the mass balance equation becomes:

$$J = Q_o C_o + sM$$

Note that all the terms still have the dimensions of amount per time.

An equation for predicting the lake concentration from loading can be produced by substituting C_{lake} for C_o (again assuming that the lake concentration is equal to the outflow concentration) and substituting $C_{\text{lake}} V$ for M :

$$J = Q_o C_{\text{lake}} + s C_{\text{lake}} V$$

Rearranging, we obtain the predictive equation:

$$C_{\text{lake}} = \frac{J}{Q_o + sV}$$

The equation can be further rearranged into the form:

$$C_{\text{lake}} = \frac{J}{Q_o} \left[\frac{1}{1 + sV / Q_o} \right]$$

With this equation, several things became clearer about loading:

- Vollenweider considers only the “total” form of the substance. He does not discriminate between dissolved and particulate forms.

- The term (J/Q_o) has the dimensions of concentration (mg/m^3) and represents the average incoming concentration of the substance (Vollenweider, 1976) assuming evaporation is minimal. This term is sometimes replaced by a symbol for incoming concentration, C_i .
- The term $(1/(1 + s(V/Q_o)))$ is really a description of the fraction of the incoming concentration that is *not* retained within the basin. In some models, retention is represented by the symbol R and the term $(1/(1 + s(V/Q_o)))$, by $(1-R)$.
- The term (V/Q_o) has the units of time and is the hydrologic residence time, T or t , which represents the average time that water remains within the lake.

Using these simpler symbols, the equation can be reduced to a simple statement of the relationship between loading and lake concentration:

$$C_{lake} = C_i \left[\frac{1}{1 + sT} \right]$$

where C_i = average inflow concentration.

Although relatively simple, the equation illustrates the major aspects of prediction with mass balance models and trophic state:

- The concentration of a substance such as phosphorus in the reservoir (C_{lake}) is directly determined by the concentration of that substance in the incoming streams (C_i). The higher the concentration in the streams entering a reservoir, the higher the nutrient concentration will be in the reservoir itself.
- Internal factors such as water residence time (T) and the net sedimentation coefficient (s) determine the amount of material that is sedimented, and therefore lost, from the water column. The longer the water residence time, the greater the amount of material that will be sedimented within the reservoir, and the lower the reservoir concentration will be.

Additional terms have been added to the equation to account for release of a nutrient from the sediments into the open water or for the biological availability of the incoming phosphorus. These additional terms can make the predictions more specific to the particular reservoir being modeled. See Chapter 9 for a discussion of these models.

D. Trophic State Classification Systems

The concept of trophic state, with its relationship of the watershed to the chemistry and biology of the water body, has become one of the primary methods of classifying lakes. Despite controversies of definition, it has endured and probably will endure because of several important reasons:

- *History and tradition.* The language and implications of trophic state are deeply ingrained in limnology. In a sense the concept of trophic state is the nutrient paradigm.

- *Communication.* When a trophic state term, such as eutrophic or eutrophication, is used, there is a general agreement as to what a lake is like in terms of nutrients and biology. This implication of interrelationships tends to communicate far more information than can be implied with the statement of the value of a single variable.
- *Education.* The trophic state concept, even in qualitative terms, is a convenient vehicle to educate the public on the simplicity, and indeed the complexity, of the relationship between land use and the biological consequences.

Trophic state classification may have begun as a continuum concept, but it rapidly evolved into a classification of “types.” Most, but not all, existing trophic classification systems, or indices, reflect this typological emphasis. The representation of this type of classification scheme is simply of list of characteristics for a specific trophic type (Table 2.1). Lakes are assigned to a trophic class based on their agreement with the characteristics on the list. This type of classification runs into difficulty when specific variables may classify the lake in different categories. This happens when the correlation between variables is not strong.

The essence of a typological trophic classification is the belief that there is a real type of lake called eutrophic in the sense that there is a real type of human called young, middle aged, or elderly. Lakes can, therefore, be classified and placed into one of these types. Eutrophication is the progressive directional change of a lake out of one type and into another. Once in a type, the lake takes on certain characteristics by which it can be recognized and, therefore, classified. Such classifications are easily recognized from lists of characteristics typical for each trophic state heading (Table 2.1).

The OECD index (Vollenweider and Kerekes, 1980) used a statistical approach to quantify the ranges of several variables within each trophic designation (Table 2.2). This index was derived by asking a group of scientists their opinion as to what was the average value for each trophic class for each variable. The summarized data were used to produce bell-shaped curves for each variable for each class (Figure 2.4). The overlap that resulted emphasized that lakes of the same concentrations may be in more than one trophic class.

The second approach to trophic classification assumes that trophic types are not real but abstractions and, to some extent, arbitrary divisions of a continuum. This approach is similar to Naumann’s original classification. In this case, the discussions have been generally along the lines of what is the appropriate trophic state variable that should be divided into trophic state classes. The discussion of appropriate variables for classification is continued in Chapter 5.

Some continuum-based classification indices emphasize that trophic state reflects a number of variables, recalling the multiple variable approach of typological schemes. For example, Huber et al. (1982) stated that “trophic state is the integrated expression of the nutritional status of a water body. As such, it is widely accepted that no single trophic indicator or parameter is adequate to completely describe and/or quantify the concept.” Multiple variable indices differ from the typological indices largely in that they quantify the multiple variables found in the trophic state list of characteristics. These approaches emphasize the collection of quantitative data and are a major advance over qualitative listings.

Table 2.1. Illustration of a Typological Trophic Classification System Based on Lake Characteristics (adapted from Rast and Lee, 1987)

Variable	General Characteristics	
	Oligotrophic	Eutrophic
Total aquatic plant production	Low	High
Number of algal species	Many	Few
Characteristic algal groups	Greens, diatoms	Blue-greens
Rooted aquatic plants	Sparse	Abundant
Oxygen in hypolimnion	Present	Absent
Characteristic fish	Deep-dwelling, cold water fish such as trout, salmon, and cisco	Surface-dwelling, warm water fish such as pike, perch, and bass; also bottom-dwellers such as catfish and carp
Water quality for domestic and industrial use	Good	Poor

Table 2.2. OECD Ranges Based on Scientists' Opinions (after Vollenweider and Carekes, 1980)

Variable	Oligotrophic	Mesotrophic	Eutrophic
Total phosphorus mean range (n)	8 3-18 (21)	27 11-96 (19)	84 16-390 (71)
Total nitrogen mean range (n)	660 310-1600 (11)	750 360-1400 (8)	1,900 390-6100 (37)
Chlorophyll <i>a</i> mean range (n)	1.7 0.3-4.5 (22)	4.7 3-11 (16)	14 2.7-78 (70)
Peak chlorophyll <i>a</i> mean range (n)	4.2 1,3-11 (6)	16 5-50 (12)	43 10-280 (46)
Secchi depth (m) mean range (n)	9.9 5.4-28 (13)	4.2 1.5-8.1 (20)	2.4 0.8-7.0 (70)

Note: Units are $\mu\text{g/l}$ (or mg/m^3), except Secchi depth; means are geometric annual means (log 10), except peak chlorophyll *a*.

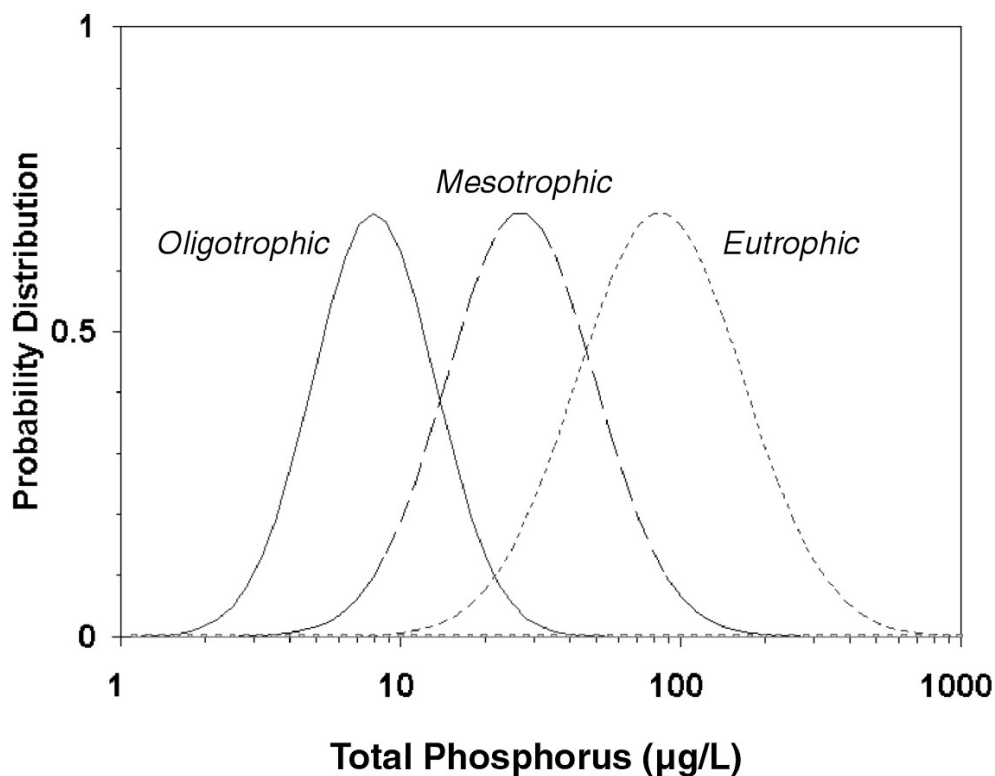


Figure 2.4. Probability distribution curves for total phosphorus by trophic status class.

Probably the most sophisticated of the multivariate indices is that of Brezonik and Shannon (1971), which uses principal components analysis to derive a trophic state index (TSI) based on seven variables: (1) TP, (2) primary production, (3) inverse of Secchi depth, (4) total organic nitrogen, (5) chlorophyll *a*, (6) specific conductance, and (7) the inverse Pearsall cation ratio ($[\text{Ca}] + [\text{Mg}] / [\text{Na}] + [\text{K}]$). Other less sophisticated indices generally combine unweighted variables by one means or another. The EPA Index (U.S. EPA, 1974) ranked lakes based on “the percentage of the 200+ lakes exceeding Lake X in that parameter”; the index was “simply the sum of the percentile ranks for each of the parameters used.” The variables used were TP, dissolved phosphorus, inorganic nitrogen, Secchi depth (500-Value [inches]), chlorophyll *a*, and minimum dissolved oxygen (15-DO_{\min}).

These multivariate quantitative indices move trophic classification from a typological concept to one assuming a continuum of values, but they suffer from several drawbacks. The indices require that all the variables be measured before an index value is derived, thus greatly increasing the cost and analytical time required. A missing value eliminates a TSI determination. Changes in a single variable often will be overlooked in the index if other indices do not change. Conversely, if index variables are correlated, then a change in one may trigger a change in a number of variables causing an exaggeration of the amount of change. Finally, a change in the index does not tell the reader what has changed; information

is lost.

Carlson (1977) suggested returning trophic state to its first principles: a quantifiable plant biomass-based concept that could fit easily into existing and future nutrient and lake models. He did not redefine trophic state but used Naumann's (1929) original idea of a classification according to plant biomass. Instead of the distinct typological classes, Carlson assumed algal biomass to be from a continuous range of values. He suggested that the commonly used trophic classes were arbitrary divisions of the biomass continuum. To emphasize the continuum nature of a biomass-based trophic state, he used a numeric rather than a nomenclatural scale, dividing the range of algal biomass based on a doubling of Secchi depth, a variable that is affected by algal density.

The original Secchi depth equation in Carlson (1977), reproduced below, illustrates how the index was constructed:

$$TSI(SD) = 10 \left[6 - \frac{\ln SD}{\ln 2} \right]$$

The basic Secchi disk index was constructed from a doubling and halving of Secchi disk transparency. The base index value is a Secchi depth of 1 m, the logarithm of which is 0.

$$\ln 1 = 0$$

$$6 - 0 = 6$$

$$10 \times 6 = 60$$

Therefore, the TSI of a 1 m Secchi depth is 60. If the Secchi depth were 2 m,

$$\ln 2 / \ln 2 = 1$$

$$6 - 1 = 5$$

$$10 \times 5 = 50$$

The index utilizes relationships between trophic variables to produce equations that allow the index to be calculated from variables other than Secchi depth. The indices for the chlorophyll and TP are derived in a similar manner, but instead of a Secchi depth value in the numerator, the empirical relationship between chlorophyll or TP and Secchi depth is given instead. For example, the TSI equation for chlorophyll is:

$$TSI(CHL) = 10 \left[6 - \frac{2.04 - 0.68 \ln CHL}{\ln 2} \right]$$

The above forms of the equations illustrate how the indices were derived, but they can be simplified for everyday use:

$$\text{TSI (SD)} = 60 - 14.41 \ln \text{SD}$$

$$\text{TSI (CHL)} = 9.81 \ln \text{CHL} + 30.6$$

$$\text{TSI (TP)} = 14.42 \ln \text{TP} + 4.15$$

The value of multiple equations is that the same TSI value should be obtained no matter what variable is used to calculate it (i.e., a common scale). This means that if data are missing for chlorophyll, for example, a similar value could be obtained from transparency.

Although these three variables should covary, they should not be averaged because neither transparency nor TP are independent estimators of trophic state. Using transparency or phosphorus as an estimator of chlorophyll is very different from assuming equal and independent status of the variables. Secchi depth and TP should be used as a surrogate, not a covariate, of chlorophyll.

In essence, this TSI scale is an indexed scale of algal biomass. Because it is directly related to lake phosphorus concentration, it fits easily into phosphorus loading models such as that of Vollenweider (1976). If a loading model can predict phosphorus concentration in the water, then the trophic state can be predicted easily as well. Work by Kratzer and Brezonik (1981) allows the index to be predicted from nitrogen concentrations as well.

$$\text{TSI(TN)} = 54.45 + 14.43 \ln(\text{TN})$$

[Nitrogen values must be in units of mg/L]

Their index could be used especially if there is any indication that nitrogen, rather than phosphorus, is limiting.

E. Uses of Trophic State Indices

Indices have several purposes. In some instances, indices take uncorrelated variables and aggregate them into a single word or value so that a general condition may be easily communicated. For example, a pollution index might include concentrations of heavy metals, pesticides, and phosphorus, which may or may not be correlated but could contribute to what the public considers to be pollution. The multivariate trophic state indices are of this type. These indices assume that trophic state consists of a number of possible attributes of lakes, ranging from nutrient concentration to hypolimnetic oxygen depletion. An index is necessary to somehow combine these various ingredients into the trophic stew and relate lakes to one another in a national continuum.

Alternatively, indices such as that of Carlson (1977) use the term “index” to mean that the variable measured is not trophic state, but an indicator of trophic state. For Carlson, trophic state is plant biomass. Chlorophyll, transparency, or even TP are variables that can estimate biomass but are really not living plant (autotroph) carbon. Even the measurement of organic carbon is not free from interferences from detritus or nonplant carbon. Trophic state is used as a surrogate for a real entity, plant biomass, that cannot be measured directly.

A third use of the term “index” that combines aspects of the first and second definition is that of simplification of a concept of measurement. For example, few readers know that the Richter scale, used to describe the magnitude of an earthquake, is the maximum deviation of a needle on a seismograph 62 miles from the epicenter. Actually, most people do not need to know the mechanics of calculating the Richter scale to have a sense of the severity of an earthquake. In the same sense, trophic state indices are shorthand methods to convey information. Total phosphorus or chlorophyll has little value in communication unless there is some standard to which the reader or listener can compare the value. In this case, saying eutrophic or TSI of 60 rather than “chlorophyll concentration is 20 µg/L” may convey information more easily because there are fewer terms to explain to an audience and fewer terms for the audience to put into the context of their own experiences.

In each of these instances, the index helps the reader equate several parameters in one indicator of enrichment condition so that an interpretation of condition is conveyed and conclusions can be made. Whether the measures of water quality compiled in a data set are used individually or collectively in an index, they are the essential, objective information the resource manager needs to determine the status of a lake or reservoir. With the representative information and a guide as benchmark criterion defining the relatively unimpaired and attainable water quality condition for comparison, the manager can classify, select, and plan for the restoration and protection of the lakes and reservoirs in his or her region.